APPENDIX J

Summary of Special Studies Supporting the EIS/EIR Impact Analysis

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Summary of Special Studies Supporting the EIS/EIR Impact Analysis

J.1 Introduction

The environmental conditions at the Salton Sea are often extreme and can be challenging for building habitat and maintaining fish and wildlife populations. The SCH Project is being designed to support shallow-water wildlife dependent on the Salton Sea (particularly fish-eating birds) and to minimize any negative impacts on wildlife or humans (from contaminants or disease vectors). The SCH Project would consist of a series of shallow ponds, several hundred acres in size and constructed on playa exposed as the Salton Sea recedes. Depending on the slope of the playa and extent of sea level decline, the ponds would have a mean depth of about 2 to 4 feet and a maximum depth of 6 feet at the outer berm. While some deeper swales would be excavated to create habitat diversity, these would not substantially affect average depths or water volume over the total pond. A range of operational scenarios have been proposed for the initial proof-of-concept phase to test which regime would best balance ecological productivity, sustainability, and potential impacts(Appendix D). Initial operations would manage the ponds as saline habitat, with salinity between 20-40 parts per thousand (ppt).

While much has been learned about the Salton Sea over the last decade, uncertainties remained for the site-specific engineering design, effects analysis, construction, and proposed operation of a restoration project. Several studies to address key uncertainties for the SCH Project were conducted for the State by researchers at the University of California Riverside (UCR) and University of California Berkeley (UCB):

1) Contaminants in water and sediment at proposed sites for SCH Alternatives

2) Hydrological and water quality modeling of SCH alternative designs and operations

3) Salinity and temperature tolerances of fish species considered for SCH ponds

4) Ecorisk modeling of potential selenium bioaccumulation

5) Selenium treatment of water supply using wetland vegetation

This document summarizes key findings of the special studies and their application to the SCH Project. For selenium ecorisk modeling study, detailed discussions are in Appendix I Selenium Management Strategies. For the remaining studies, the study approach and results to date are summarized below from the researchers' reports. The SCH Project team then evaluated the potential implications and application to the SCH Project, which have been considered in the proposed design, operations, and impact analysis.

J.2 Contaminants in Water and Sediment

J.2.1 Purpose and Need

The SCH Project ponds would be constructed on recently exposed or soon-to-be exposed playa, and supplied with water blended from the Salton Sea and either New River or Alamo River. One issue is potential toxicity impacts from contaminants in sediments or water at the proposed SCH ponds. Water quality in the Salton Sea and its tributaries is influenced primarily by the quality of Colorado River water imported into the watershed and land use activities, principally agriculture, that contribute salts and other constituents to the Salton Sea inflows (DWR and DFG 2007). Some of those constituents, such as selenium, may contribute to toxicity risks to the ecosystem and humans through accumulation in the sediment and cycling through the food web. Sediment and water samples were collected from the alternative SCH sites and tested for contaminants.

J.2.2 Approach

Chris Amrhein and colleagues at UCR collected samples of sediment and water in summer 2010 within the footprint of the proposed alternative sites at the New and Alamo rivers. Sediment samples were collected from the surface (0-5 cm), as well as subsurface (5-15 cm deep and 15-30 cm deep) to look at historic deposition. Sediment samples were taken from exposed playa sediments and from submerged sediments, although submerged samples were not evenly collected across the potential pond sites. The samples were tested for phosphorus, trace metals and metalloids (selenium, boron, arsenic), PCBs, and organochlorine insecticides (including DDT), pyrethroid insecticides, organophosphorus insecticides, and other contaminants (Amrhein and Smith 2011; Wang et al. 2011).

An experiment was conducted to examine the release of selenium from sediments (Amrhein et al. 2011). Selenium is often present in reduced forms (less bioavailable and therefore less toxic) when wetlands are submerged and have high organic matter. When the water level is lowered, selenium can become oxidized and more bioavailable. The initial wetting period could increase selenium bioavailability by allowing solubilization of oxidized selenium into the water (DWR and DFG 2007). This experiment was designed to represent a worst-case scenario where a relatively high concentration of sediment (50:50 wet sediment to water) is mixed into the overlying, aerobic water and selenium is oxidized. These samples were incubated for up to 235 days with well-aerated water at salinities 21 ppt and 13.7 ppt. Water was periodically decanted from samples and selenium concentrations measured.

The researchers also evaluated the relationship between aeration time due to the receding Salton Sea shoreline and soluble selenium in sediment pore water. Samples were collected from three areas in Red Hill Bay at varying distances from the shoreline. The researchers estimated the “time exposed” based on the distance to the water, the slope of the land, and the elevation of the Sea over time.

J.2.3 Results

Selenium

Mean water selenium concentrations were 1.2 micrograms per liter (μg/L) in Salton Sea, 1.8 μg/L in the New River, and 4.1 μg/L in Alamo River (Amrhein and Smith 2011). Mean sediment selenium concentrations at proposed Project sites were 1.1 milligrams per kilogram dry weight (mg/kg dw) (range 0.54–2.3 mg/kg dw). The majority of sediment samples (63 percent) were less than 1 mg/kg dw and would be considered “low risk.” The remaining 37 percent of the samples were between 1 and 4 mg/kg dw (only two samples exceeded 2.5 mg/kg dw) and were considered in the “level of concern” category. No sample exceeded the “toxicity threshold” value of 4 mg/kg dw.

The solubilization data indicate that oxidation due to draining and aeration of the sediments, as the Sea recedes, can increase the water-soluble selenium (Amrhein et al. 2011). Mean water selenium concentrations after 131 days incubation were 6.5 - 8.2 μg/L at Alamo River playa sites (n=15), 11.9 μg/L in Alamo River, and 12.8 at New River playa (Table J-1). Cumulative release of sediment selenium ranged from 18.9 μg/kg (8.1 percent of total sediment selenium) after 194 days in Morton Bay, up to 48.8 μg/kg (37 percent of total sediment selenium) after 235 days) in the Alamo River channel (Table J-1). The rate of release was mostly decreasing over time, suggesting the sediments will be a decreasing source of selenium.

Table J-1 Selenium Released from Oxidized Sediments

Location	Samples	Total Sediment Selenium (mg/kg)	Water Selenium (µg/L) after 131 Days	Cumulative Selenium Released from Sediments (µg/kg and Percent Oxidized)	
				194 Days Incubation	235 Days Incubation
Alamo River	2	0.18	11.9	--	48.8 (36.9%)
Alamo River - Red Hill Bay	6	1.26	6.5	--	27.9 (4.5 %)
Alamo River - Delta	4	0.36	8.2	19.6 (8.0 %)	--
Alamo River - North Morton Bay	5	0.46	6.8	18.9 (8.1%)	--
New River Bay	2	0.23	12.8	29.9 (14.9%)	--

Source: Amrhein et al. 2011 (mean values reported)

Anaerobic conditions in the sediments result in very low selenium concentrations because reduced forms of selenium have the lowest solubility. Sediment selenium concentrations were positively related to organic carbon, but the oxidation rates and amount released into water did not appear affected by carbon content, salinity, location, or depth of sample core. Rather, the release of selenium appeared controlled by the amount of oxidizable iron present in sediments. The amount of released was most strongly linked to presence of oxidizable iron (Fe [III]), which adsorbs selenium (in the form of selenite) in the sediment, resulting in less selenium dissolving into the water.

Selenium concentrations were also measured along a transect representing sediments that are currently flooded, drained for approximately 1 month, and drained approximately 2 months due to the receding Salton Sea. Water-soluble selenium concentrations were twice as high from sites drained 1 month (approximately 4 µg/L) and three to four times higher from sediments drained two months (approximately 6-8 µg/L), compared to flooded sites (approximately 2 µg/L).

Amrhein calculated the amount of selenium potentially released to the overlying water in a pond system, assuming pond sediments were aerobic to a depth of 5 cm, the overlying water averaged 1 meter deep with no water exchanges, wet bulk density of the sediments 1.8 g/cm3, and 10 µg/L selenium (85th percentile of all water samples). Based on these assumptions, the contribution from the sediments would increase the selenium in the overlying water by 0.9 µg/L (C. Amrhein, personal communication 2011). This is a conservative estimate, since water would be exchanged in the SCH ponds at a rate dependent on flow operations (likely range of residence times 4 to 32 weeks) (Appendix D).

In conclusion, aerated conditions can produce oxidized selenium, which is more soluble, although the amount dissolved into water will depend on several factors, most particularly the presence of iron (Fe [III]). This suggests an initial “flush” of selenium from the sediments could occur and is consistent with observations at the Reclamation/USGS Saline Habitat Ponds (Miles et al. 2009). However, dissolved selenium in inflow water would likely pose a greater relative risk to wildlife bioaccumulation than selenium released from sediment (Amrhein et al. 2011).

Pesticides

Levels of chlorinated insecticides and pyrethroids were measured in water of the New and Alamo rivers and in the bed sediments at potential SCH pond sites (Wang et al. 2011). In the water (four samples per river), most organochlorine pesticides were below 1.5 nanograms per liter (ng/L) or were not detected. Chlorpyrifos was the most frequently detected, but only one sample at the New River was elevated (80 ng/L). The most commonly detected pyrethroid was permethrin (3.3-7.5 ng/L) with fenpropathrin detected once at elevated levels (New River, 11.6 ng/L). The number of samples was deemed too small to allow concrete conclusions about ongoing contributions of pesticides to the SCH ponds (Wang et al. 2011).

Sediment samples were taken at three depths (0-5 centimeters [cm], 5-15 cm, and 15-30 cm below the surface) in order to discriminate potential differences in deposition of legacy (i.e., organochlorines) and current-use pesticides (i.e., pyrethroids). Total sediment pesticide concentrations detected ranged from 0.2 to 120 nanograms per gram [ng/g]. Sediment pesticide concentrations, particularly organochlorines, were greatest at the mouth of both the New and Alamo rivers. Dichlorodiphenyltrichloroethane (DDT) and its metabolites were detected in all samples, and dichlorodiphenyldichloroethylene (DDE) was the predominant pesticide residue. In general, the concentrations of organochlorine pesticides were higher in the 5–30 cm depth interval than in the 0–5 cm depth interval (more recent deposition). This pattern correlates with the banning of most organochlorine pesticides, including DDT, in the United States in the 1970s. Mean DDE concentrations in air-exposed sediments at 0-5 cm deep and 15-30 cm deep were 2.6 ng/g surface and 10.9 ng/g subsurface at New River sites, and 12.1 ng/g surface and 25.5 ng/g subsurface at Alamo River sites. Organochlorine pesticide concentrations showed a pattern of decreasing concentration with distance from the river mouths. The highest DDE concentrations were found immediately adjacent to the Alamo River mouth in Morton Bay and in New River East. Lower concentrations of DDE were found at the Alamo River-Davis Road (north of Morton Bay) and New River Middle sites. The lowest DDE concentrations were found at the New River Far West sites. This spatial pattern is consistent with the overall circulation pattern in the Salton Sea, which tends to move counterclockwise.

The submerged samples typically had lower DDE concentrations than air-exposed sediments (Wang et al. 2011). The researchers hypothesized that this could be due to more extensive degradation in the submerged areas under reduced conditions. However, this could be an artifact of uneven sampling distribution. The samples from Red Hill Bay (southwest side of Alamo River) and Morton Bay (northeast side of Alamo River) were grouped into a single “Alamo River - Red Hill” region. All the submerged samples were from Red Hill Bay, which is upcurrent of the prevailing circulation that would carry river-borne sediment toward Morton Bay and northward.

A screening criterion of 31.3 ng/g DDE was identified as a Probable Effects Concentration (PEC) for general ecotoxicity, based on sediment guidelines developed by MacDonald and others (2000) and suggested by the Colorado River Basin Regional Water Quality Control Board (CRBRWQCB 2010) to prevent direct toxicity to the macroinvertebrate population, which serves as a food base for fish and insectivorous birds. The frequency of surface sediment samples exceeding this guideline was 18 percent at Alamo River-Morton Bay (32.41 ng/g maximum); 14 percent at Alamo River-Davis Road (34.40 ng/g maximum); and none at New River sites. The frequency of subsurface samples exceeding the PEC was 37 percent at Alamo River-Morton Bay (102.60 ng/g maximum); 7 percent at Alamo River-Davis Road (38.26 ng/g maximum); and 10 percent at New River East (41.16 ng/g maximum); 3 percent at New River Middle (33.51 ng/g maximum); and none at New River West.

Chlordane (organochlorine, < 3 ng/g Alamo River, < 1.2 ng/g New River) and bifenthrin (pyrethroid, < 1.9 ng/g Alamo River, < 0.5 ng/g New River) were also detected, but at lower levels than DDE. Other pesticides were infrequently detected (Wang et al. 2011). It is worth noting that bifenthrin, a pesticide first registered for use in the late 1980s -- early 1990s, also increased concentrations with depth, which could indicate that the deepest sediments sampled in the study represent relatively young sediments (personal communication, J. Orlando 2011).

J.2.4 Application to SCH Project

Selenium

The relative pattern of water selenium concentrations showed highest concentrations in the Alamo River, then the New River, and lowest in the Salton Sea. Although concentrations measured by Amrhein and Smith

(2011) were slightly lower than those reported by the U.S. Bureau of Reclamation (C. Holdren, Reclamation, unpublished data, quarterly sampling 2004-2010), the pattern is consistent. Therefore, options to reduce selenium inputs would include operating the SCH ponds with New River water instead of Alamo River and/or operating the ponds at higher salinities (i.e., less river water and more Salton Sea water).

The solubilization experiment suggests that an initial "flush" of selenium released from the rewetted sediments could occur. Selenium solubilization from sediments would be temporary and would decline over time. Reducing water retention time and increasing flow-through of the ponds for several weeks or months following initial filling could be used to flush soluble selenium from the ponds (Amrhein et al. 2011). The volume of dissolved selenium from inflow water would likely pose a greater relative risk to wildlife bioaccumulation than selenium released from sediment.

Pesticides

To apply these data to the current SCH Project alternatives, mean DDE concentrations were recalculated from the raw data in Wang and others (2011) by combining air-exposed and submerged samples into geographic categories that matched the SCH Project alternatives (Red Hill Bay samples southwest of Alamo River were excluded because this area is no longer under consideration for Alternatives 4-6). Also, nondetects or undetected levels of DDE were defined as 0.01 ng/g for purposes of avoiding zeroes and allowing those extremely low values to be reflected in the means (Table J-2).

Table J-2 DDE Concentrations in Sediment at SCH Project Area (ng/g)

Location	Surface Mean (# samples)	Surface Maximum	Subsurface Mean (# samples)	Subsurface Maximum
New River - East	6.52 (11)	23.71	9.10 (21)	41.16
New River - Middle	2.78 (15)	7.99	5.44 (29)	33.51
New River - Far West	1.14 (6)	2.90	0.89 (13)	2.41
Alamo River - Morton Bay	13.66 (11)	32.41	25.02 (19)	102.60
Alamo River - North (Davis Road)	13.41 (7)	34.40	9.16 (14)	38.26

Source: Calculated from raw data in Wang et al. 2011. Surface (0-5 cm deep) and subsurface (5-15 cm and 15-30 cm deep). Nondetect values were defined as 0.01 ng/g for purpose of calculating means. Samples were pooled for air-exposed and submerged sites within each location.

Mean DDE concentrations in sediments at New River were 1.14 to 6.52 ng/g at the surface (0 to 5 cm deep) and 0.89 to 9.10 ng/g subsurface (5 to 15 cm and 15 to 30 cm deep). Mean DDE concentrations in sediments at Alamo River were 13.41 to 13.66 ng/g at the surface (0 to 5 cm deep) and 9.16 to 25.02 ng/g subsurface (5 to 15 cm and 15 to 30 cm deep) (Table J-2). Current DDE concentrations in surface sediments (0 to 5 cm deep) represent undisturbed existing conditions. For comparison, mean sediment DDE levels were measured at nearby sites (0-5 cm deep) by USGS in 2006-2008 (Miles et al. 2009): 4-48 ng/g at their saline habitat ponds (SHP), 41-56 ng/g in Alamo River, 15-41 ng/g in the Salton Sea near Alamo River, 60-98 ng/g at the Freshwater Marsh near Morton Bay, and 2-6 ng/g at the D-Pond on the Sonny Bono Salton Sea National Wildlife Refuge (NWR) (Miles et al. 2009). With the exception of the D-Pond, these concentrations are similar or higher than the levels measured at the SCH alternative sites.

Exposure to the more contaminated subsurface sediments would occur only in those areas disturbed by excavation for berms, swales, and islands, and would be averaged across the entire pond area including undisturbed areas. Therefore, expected DDE concentrations were calculated for each SCH alternative, based on field measurements of surface sediments (0 to 5 cms) and subsurface sediments (5 to15 and 15 to 30 cm deep) (Wang et al. 2011), and weighted according to proportion of pond area that would remain undisturbed but inundated (surface 0- to 5-cm concentrations) and area disturbed by construction [borrow ditches for berms, excavated swales and channels, borrow for habitat islands) (subsurface 5- to 30-cm concentrations)]. "Mean" is the area weighted average calculated using mean values for surface and subsurface sediment. Because DDE concentrations below 30 cm are unknown and construction could disturb deeper sediments, hypothetical "maximum" concentrations were also calculated using maximum observed values of surface and subsurface sediments, as a hypothetical upper bound of potential risk (Table J-3). The incremental increase in DDE concentration across the pond unit compared to existing levels was minor.

Table J-3 Area-Weighted Mean Sediment DDE Concentrations (ng/g) for Existing Conditions and SCH Project Alternatives

		Existing Conditions[1]		SCH Project	
		Estimated for Undisturbed Surface Sediments		Estimated for Constructed Ponds	
River	Pond units	Calculated from mean	Calculated from maximum	Calculated from mean	Calculated from maximum
New River	New East	6.5	23.7	7.1	27.6
	New Middle	2.8	8.0	3.6	15.7
	New Far West	1.1	2.9	1.0	2.7
Alamo River	Alamo Morton Bay	13.7	32.4	15.7	45.0
	Alamo - north (Davis Road)	13.4	34.4	12.9	34.8

1. DDE concentrations (mean and maximum values) in undisturbed surface sediments (0 to 5 cm deep) measured at each location (Amrhein and Smith 2011; Wang et al. 2011)

Because the concentrations of DDE and bifenthrin increased with depth sampled, it is possible that deeper sediments potentially exposed during SCH construction (excavation of playa sediments for berms and islands) could contain higher concentrations of organochlorine pesticides than reported by Wang et al. (2011). The fact that a current use pesticide like bifenthrin also increased with depth could indicate that the deepest sediments sampled could represent relatively young sediments (personal communication, J. Orlando 2011). Also, concentrations of DDE in suspended sediments collected from the Alamo River and New River in 2006-07 (Orlando et al. 2008) are comparable to concentrations seen in bed sediments in this study, suggesting that the current influx of DDE (and likely current-use pesticides) associated with suspended sediments to the Salton Sea may be of concern with respect to SCH construction and operations (personal communication, J. Orlando 2011). Targeted sampling of sites that would be actually be disturbed by construction may be warranted.

J.3 Hydrological Modeling

J.3.1 Purpose and Need

To provide suitable habitat, a shallow water system should maintain stable water balance, well-oxygenated conditions, productive food web, suitable salinity and temperature for fishery resources, limited resuspension of sediments, and flexible management practices. Salinity is an important water quality parameter that would be managed to maximize biological productivity and minimize adverse effects from water quality constituents (i.e., selenium loading, bioaccumulation through emergent vegetation) and other factors (vector control).

Options considered for establishing a salinity gradient in ponds include evapoconcentration of salts as water flows through the ponds, or blending river water with saltwater. The inflow to the SCH ponds would be a blend of nutrient-rich agricultural runoff (New or Alamo Rivers) and Salton Sea water. This has the potential for high algal production, anoxic conditions, and accumulations of ammonia and sulfide. Finally, because the shallow ponds would be located in a desert environment, water temperatures would range widely both seasonally and diurnally.

The purpose of this special study was to inform the engineering design and operational guidelines by addressing several key questions. First, what is the most effective means to achieve the desired salinity range for the ponds? Second, would the expected pond design and operations result in water quality conditions that could support a productive fish community and therefore meet project goals (support fish-eating birds)? Finally, are there particular periods or situations where conditions could exceed biological tolerances?

Hydrologic modeling by Barbara Barry and Michael Anderson (UCR) was used to explore how different potential pond configurations, source waters, and water operations could affect the expected physical, chemical, and biologic conditions in SCH ponds. This analysis involved successive iterations between UCR and the SCH Project design team to refine design alternatives and model parameters.

J.3.2 Approach and Results

The SCH Project would consist of a series of shallow ponds, several hundred acres in size and constructed on playa exposed as the Salton Sea recedes. Pond design parameters included depth, morphometry (pond shape, which affects water volume), and fetch (potential for wind mixing). Operational parameters included hydraulic residence time (4 and 16 weeks), source water (New River, Alamo River, and Salton Sea), and influent salinity.

UCR applied two models to simulate the physical, chemical, and ecological conditions in the SCH ponds. The first modeling exercise examined the water and salt balance of two pond designs: (1) interconnected ponds with flow cascading serially from one to another downslope (“sequential” ponds), and (2) independent ponds each receiving direct delivery of input water (“concurrent” ponds). Water column temperatures and salinities were predicted by DYRESM, a 1-dimensional thermodynamic-hydrodynamic model that uses meteorological data (2006-2008) combined with basin characteristics, hydrological inputs and outflows, and influent salinity and temperature. The second modeling exercise predicted vertical profiles of water temperature and dissolved oxygen (DO) for different pond designs and operations. This analysis used the Computational Aquatic Ecosystem Dynamics Model (CAEDYM), a 1-dimensional model that uses DYRESM outputs to model a wide range of water quality conditions (temperature, DO, nutrients, chlorophyll) and biological conditions (phytoplankton, zooplankton and fish).

Blending Sea and river water is the only feasible means to achieve the desired salinity range (20-40 ppt) across all ponds. Evaporation would increase salinity over time, depending on mean depth (indicative of water volume) and residence time. With an inflow salinity of 20 ppt and hydraulic residence time of 60 days, the resulting pond salinity would be 30 ppt in a 0.5 m deep pond and 23 ppt in a 1.5 m deep pond. However, relying solely on evapoconcentration of river water (2 ppt) would never achieve target salinities, and would increase selenium loading to ponds because water selenium concentrations are greater in the rivers than the Salton Sea.

The water quality modeling provided one-dimensional vertical profiles of temperature and DO, hourly over a three-year simulation period. Temperature profiles were very similar across scenarios. Water temperatures would periodically drop below tilapia tolerances (11-13°C [52-55°F]) during December through February. Thermal stratification occurred in ponds with smaller surface area (200 acres), which have less fetch and therefore less wind mixing, than larger pond areas. Deeper ponds (1.5 m mean depth) would experience stratification more frequently than shallower ponds (0.76 m mean depth).

Nutrient concentrations are high in the New and Alamo rivers due to contributions from agricultural runoff. Elevated nutrients would produce eutrophic conditions and algal blooms that could lead to anoxia. Modeling results suggested that ponds would become stratified in summer (May-October). Bottom waters would experience anoxia, particularly during periods of algal blooms in spring (March-May) and fall (October). Depending on the pond scenario, increasing residence time (ranging from 4 weeks to 32 weeks) had no effect or increased somewhat the frequency of anoxia. River source (New or Alamo) for blended water supply had little effect on stratification or anoxia. Phytoplankton was more abundant with Alamo River blended water. Zooplankton did better with New River blended water and consequently reduced phytoplankton slightly.

J.3.3 Application to SCH Project

In general, this 1-D modeling validated the conceptual understanding of how these ponds would function. While the models are not sufficiently site specific or complex to truly answer questions of pond sustainability, they did highlight some issues for consideration.

The most effective means of achieving the desired salinity range for the ponds would be blending sea and river water, not evapoconcentration. Salinity within a pond would increase over time due to high evaporative losses in this climate (7-10 ppt increase with a 60 day residence time), which would require additional input of river water to offset to maintain a target salinity. If a sequential pond design is used (water flowing through a series of ponds), then a salinity gradient increasing from first to last ponds would be expected.

The models, as limited as they are, confirmed assumptions that a productive aquatic system could be developed that would include fish for birds. This exercise proved useful to look for trends and periods of concern. Stressful conditions would occur periodically. Water temperatures would be too cold for tilapia to tolerate for periods during December to February. Anoxia would occur near the bottom and occasionally complete anoxia through the water column when phytoplankton blooms occur in spring and fall. Stratification would maintain a layer of oxygenated water near the surface. Bottom anoxia is more of a concern for benthic invertebrates than for tilapia, which can tolerate conditions of 1 μg/L DO and can move upwards to oxygenated water near the surface. Model results have guided development of the proposed operations and have focused the number of operational scenarios to be validated in the proof-of-concept phase (Appendix D).

J.4 Fish Tolerance

J.4.1 Purpose and Need

The fish species that would be stocked in the ponds would have to survive and reproduce given the expected water quality conditions, both managed (salinity) and uncontrolled (air temperature, wind mixing, dissolved oxygen). Tilapia appear to meet many of the requirements for a productive, sustainable fishery resource for piscivorous birds (DFG 2011). Tilapia are currently in the Salton Sea, are an important forage species for birds, and have impressively wide tolerances for salinity (currently persisting in the Sea at 53 ppt) and low dissolved oxygen. Their main drawback, other than potential competition with desert pupfish, is whether they could handle the lowest water temperatures predicted for SCH ponds. While the SCH ponds could be operated to adjust salinity (proposed range 20-40 ppt, Appendix D), it will be difficult if not impossible to control water temperatures that naturally fluctuate widely in this desert climate.

This laboratory experiment by Dan Schlenk and Varenka Lorenzi of UCR tested the survival tolerances of different tilapia species exposed to various combinations of salinity and temperature in order to inform design of operational scenarios and selection of fish species for stocking.

J.4.2 Approach and Results

Among the fish that currently live in the Salton Sea area, three forms of tilapia (*Cichlidae, Perciformes*) have been identified as potential candidates to stock the SCH ponds (DFG 2011): California Mozambique hybrid tilapia *Oreochromis mossambicus x O. urolepis hornorum* ("Mozambique hybrid tilapia"), an unidentified species resembling blue tilapia *Oreochromis aureus,* and redbelly tilapia *Tilapia zillii*. Blue are considered more cold tolerant than other tilapia species in general (Popma and Masser 1999). In addition, Mozambique hybrid tilapia raised in aquaculture were also considered because of its availability from local hatcheries, in anticipation of the wild stocks in the Salton Sea eventually failing with increasing salinity.

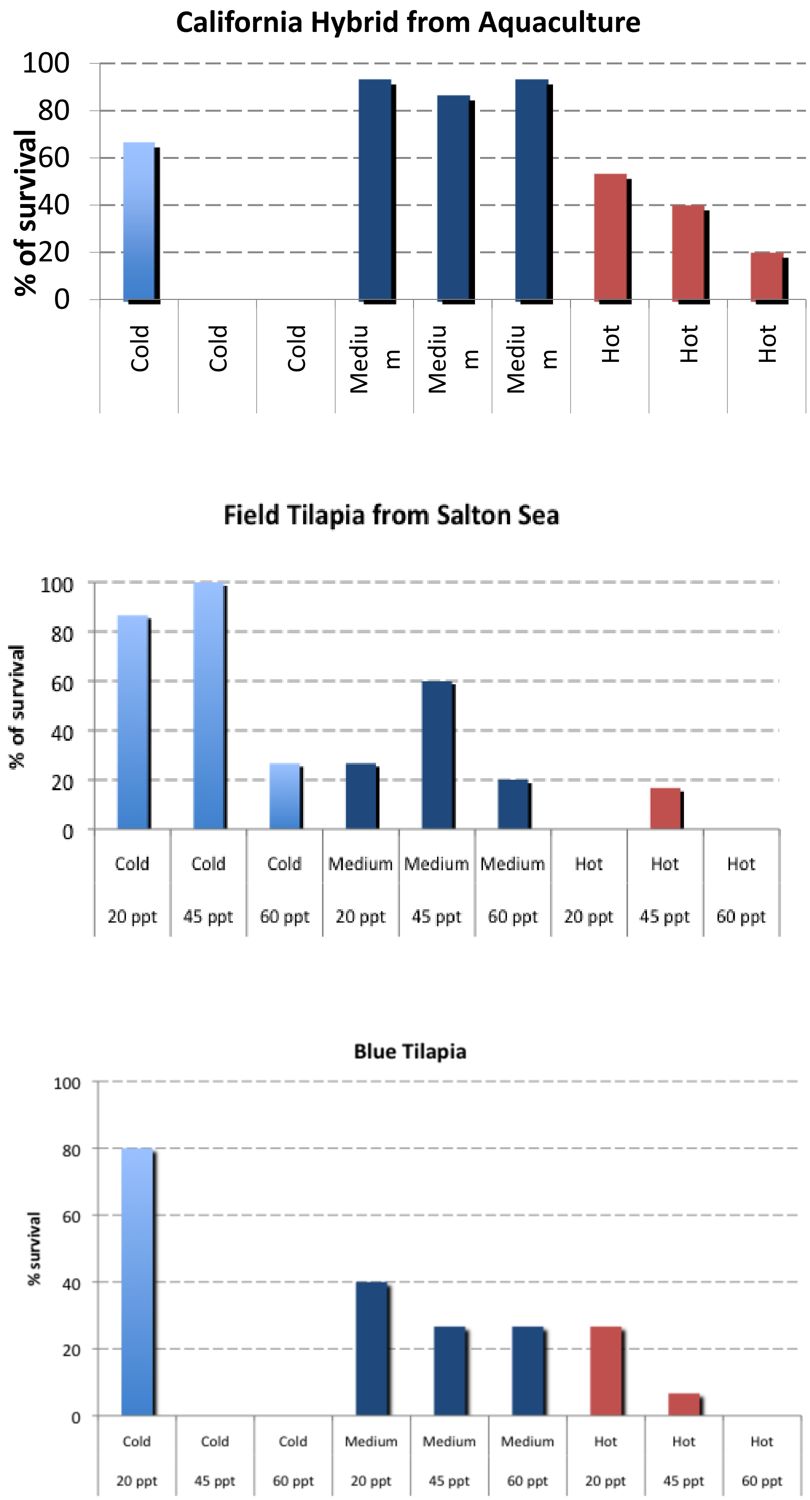
The tested fish included Mozambique hybrid tilapia (two strains: wild fish from Salton Sea and an aquaculture strain from a local fish farm), fish from a blue tilapia assemblage in the New River ("New River blue tilapia"), and redbelly tilapia collected from an agricultural drain at the northeast Salton Sea (Lorenzi and Schlenk in preparation). Juvenile fish were collected, acclimated in the lab, and then exposed to different combinations of salinity and temperature. The three salinity concentrations (20, 45, and 60 ppt) were obtained by blending water from the Salton Sea and New River, similar to the approach that would be used to operate the SCH ponds. The three temperature regimes mimicked daily fluctuation of 5 degrees Celsius (°C): cold 11-16°C (52-61 degrees Fahrenheit [°F]), warm 23-28°C (73-82 °F), and hot 33-38°C (91-100°F). After an acclimation period, survival and condition of fish was tested over a 30-day period.

When maintained at 20 ppt salinity, the New River blue tilapia had the best overall survival across all temperature regimes (80 percent survival at cold, 40 percent at warm, and 27 percent at hot) (Lorenzi and Schlenk in preparation). Redbelly tilapia survival was very poor in the lab, but this likely was due to other stressful conditions in captivity, namely aggression. It does not appear appropriate to draw conclusions about this species' thermal and salinity tolerances from such data. While most strains and species had moderately good survival in 45 ppt and 60 ppt conditions at warm temperatures, all species showed poor survival in hot high-salinity (60 ppt) conditions.

In the cold treatment (11-16°C), the fishes were less active and fed less. The Mozambique hybrid collected from the Sea had the best overall survival at cold temperatures, with excellent survival at 20 ppt (100%) and 45 ppt (85 percent), and even some survival at 60 ppt (27 percent) (Figure J-1). The California Mozambique hybrid from aquaculture (67 percent) and the blue tilapia (80 percent) were able to survive only when the salinity was low (20 ppt), indicating that the cold temperature represents a stressor for osmoregulation. Surprisingly, the New River blue tilapia did not have better survival than Mozambique tilapia in cold conditions.

In the warm treatment (23 - 28°C), some individuals in all four species and strains of tilapia managed to tolerate salinities up to 60ppt. Remarkably, some of the blue and redbelly tilapia also survived these extreme salinities, thus demonstrating the broad osmoregulatory ability typical of tilapia in general, even in these two species typically found in freshwater. At medium temperatures California Mozambique hybrid from aquaculture showed the best survival at all salinities (85-90 percent), while the wild type did well only at 45ppt. This salinity is the closest to current Sea salinity (51 ppt), so these fish were probably best adapted to osmoregulate at this salinity.

At hot temperatures (33 - 38°C), all fishes showed sign of stress and the final survival rate was quite low. The California hybrid from aquaculture did best overall and in particular at 20 ppt salinity. Only 17 percent of the California hybrid from the field survived, and only at 45 ppt salinity.



Source: Lorenzi and Schlenk (in preparation)

Figure J-1 Survival Rates of Tilapia (Aquaculture and Wild Strains of California Mozambique Hybrid Tilapia, and New River Blue Tilapia)

J.4.3 Application to SCH Project

Stocking different tilapia species or strains (individually or in combination) among the SCH ponds could be employed to increase enhance stability of the fishery resource in the ponds in the face of seasonal and annual fluctuations in water quality parameters. The Mozambique hybrid tilapia seemed to be the most resistant species across all treatments. The wild-type from the Salton Sea was most likely to survive the cold, and the aquaculture-type is the most likely to survive at high and medium temperatures. The New River blue tilapia also had good survival in cold, but only when salinities are lower (20 ppt). Redbelly tilapia are still candidates, because their poor experimental survival appeared to be due in part to lab conditions.

These results also suggest that pond operations should be adjusted to maintain lower salinities during the winter, when cold temperatures stress fish and presumably reduce osmoregulatory abilities and tolerance. Such seasonal variation in pond salinity regime has been incorporated in proposed operational scenarios (Appendix D).

J.5 Selenium Ecorisk Modeling

J.5.1 Purpose and Need

Selenium in river water supplying the SCH ponds could bioaccumulate through the food web (discussed in detail in Appendix J). The most serious toxic impacts of selenium manifest themselves in bird reproduction, namely reduced hatchability of eggs and embryo deformities (Ohlendorf and Heinz 2011). Selenium ecorisk modeling was conducted by James Sickman and colleagues at UCR to evaluate the potential risk of transfer and bioaccumulation in the foodweb under different SCH alternatives and operational scenarios (Sickman et al. 2011).

J.5.2 Approach and Results

Sickman et al. (2011) used the progressive modeling approach of Presser and Luoma (2010) to simulate transformation of dissolved selenium into particulate organic matter and selenium bioaccumulation in invertebrates, fish and birds. Since reproductive effects in birds are the most sensitive indicator of selenium toxicity (Ohlendorf and Heinz 2010), the assessment end-point was egg selenium concentration. In bird eggs, 6 µg/g dw is a conservative and widely reported toxicity reference value (Ohlendorf and Heinz 2011). The responses to selenium vary among bird species, ranging from “sensitive” (mallard) to “average” (stilt), and “tolerant” (avocet) (Skorupa 1998, as cited in Ohlendorf and Heinz 2010). Risk of impaired reproduction (reduced hatching success) can start to occur at egg concentrations of 6-12 µg/g dw. The risk of teratogenesis (deformed embryos) starts to occur above 12 µg/g dw for sensitive species, and above 20 µg/g dw for moderately sensitive species (Ohlendorf and Heinz 2010).

The model tested different operational parameters, including New or Alamo River water blended with Salton Sea water to achieve operational salinity of 20 ppt or 35 ppt, and a worst case future scenario of only river water (water selenium concentration up to 10 µg/L).

Overall, model results suggest that fish and bird eggs in SCH ponds utilizing Alamo River water would have about 50 percent higher selenium concentration compared to SCH ponds utilizing New River water (Table J-4). This is due to higher dissolved selenium levels in the Alamo River water relative to the New River. Similarly, risk increases as salinity decreases, with about 25-30 percent higher selenium concentrations predicted at a salinity of 20 ppt relative to 35 ppt. Further details on various model scenarios and results are provided in Appendix I.

Table J-4 Modeled Selenium Concentrations in Biota

River Source	Salinity	Water (µg/L)	Macro Invertebrates	Fish (Whole)	Bird Eggs (Invertebrate Eaters)	Bird Eggs (Fish Eaters)
New River	20 ppt	2.6	4.2	5.5	7.6	8.3
	35 ppt	2.0	3.3	4.3	6.0	6.5
Alamo River	20 ppt	4.0	6.6	8.5	11.6	12.7
	35 ppt	2.8	4.5	5.9	8.1	8.9

Selenium concentrations in biota = micrograms per gram dry weight (µg/g dw).
Source: Sickman et al. 2011 (General Model simulation)

J.5.3 Application to SCH Project

The modeling results yield several findings with relevance to SCH design and operation. First, the selenium risk in SCH ponds constructed with Alamo River water would likely be substantially higher than in ponds utilizing New River water. Risk characterization indices suggest there would be moderate to high risk for reduced egg viability in black-necked stilts in Alamo River SCH ponds and that the risks would be elevated above current risk levels. Second, inverse modeling supports the premise that higher salinity levels would result in lower risk from selenium. Salinity of 35 ppt is recommended to reduce risk of reproductive effects (< 6 µg/g dw). If low to moderate levels of reduced hatching success are deemed acceptable, then salinity levels closer to 20 ppt would be adequate for New River SCH ponds.

The actual magnitude of selenium impacts for the implemented Project could be lower than modeled. First, the actual concentrations could be lower because birds' foraging range would likely extend beyond the SCH ponds to include other habitats that have lower selenium levels (i.e., freshwater ponds at the Sonny Bono Refuge). Second, when the model was run using parameters estimated from the SHP complex, the modeled egg selenium concentrations were greater than the actual measured egg concentrations (Miles et al. 2009), indicating that this ecorisk model is a very conservative estimator of risk.

J.6 Selenium Treatment by Wetland Vegetation

J.6.1 Purpose and Need

One approach to reducing selenium risk to wildlife would be treating the river water supplying the SCH ponds to reduce water selenium concentrations. Only river water would need to be treated, since Salton Sea water is less than 2 µg/L. Biological treatment, such as constructed wetlands or algal treatment, appears to have the most applicability, although there is lack of consensus among experts and in the literature (Cardno ENTRIX 2010). In the New River, the constructed Imperial and Brawley Wetlands were designed to reduce nutrients as well as selenium (Johnson et al. 2009). A key uncertainty is whether constructed wetlands could reliably reduce water selenium concentrations to less than 5 µg/L (CRBRWQCB 2006) or even 2 µg/L.

A study currently underway by Norman Terry of UC Berkeley is evaluating the effectiveness of using a water treatment system that incorporates constructed wetlands to manage selenium. Phytoremediation (biological treatment by wetland plants and the microbial community they support) is a potential technique to reduce selenium. The removal of selenium by biological volatilization to the atmosphere is highly desirable because it leads to a net loss from the aquatic system, thereby preventing its entry into the food chain.

J.6.2 Approach and Interim Results

This study is investigating approaches to enhance volatilization (Lin and Terry 2003), either by selecting wetland plant species that are more effective at volatilization, or by adding a carbon source (e.g., molasses) to stimulate bacterial processes and thus enhance volatilization. Criteria for selecting plants include ability to sequester or volatilize selenium, rapid growth and spread, and suitability for the Salton Sea climate and habitat. Preliminary results from laboratory mesocosm experiments suggest that different wetland designs and management techniques have the potential to reduce selenium concentrations to levels substantially lower than 5 μg/L.

The next phase of the work will include building a pilot constructed wetland water treatment system in the south Salton Sea area to see if laboratory results could be transferred into the field. In addition, further monitoring of selenium removal is planned for the Brawley and Imperial constructed wetlands.

J.6.3 Application to SCH Project

The SCH ponds would be managed through a combination of source control and pond management to reduce selenium exposure and risk to biota, depending on the alternative chosen and project operations (Appendix I). The levels of selenium in the water, sediment, and bird eggs from the ponds would be monitored. If these measures do not reduce or mitigate risk to acceptable levels, it may be necessary to consider water treatment techniques as part of adaptive management. However, water treatment would not be implemented as part of the SCH Project.

In the future, as the Salton Sea becomes more saline, water treatment to remove selenium may become necessary as more river water is used to maintain suitable salinities for the fish community. More information about performance and feasibility of biological treatment techniques would be required to determine whether this would be an appropriate selenium control measure at a future phase of SCH Project implementation. This set of studies currently underway would refine understanding of constructed treatment wetlands.

J.7 References

Amrhein, C. and W. Smith. 2011. Survey of selenium, arsenic, boron and pesticides in sediments at prospective SCH sites. Report prepared by University of California Riverside for the California Department of Water Resources. January 20.

Amrhein, C., W. Smith, and W. McLaren. 2011. Solubilization of selenium from Salton Sea sediments under aerobic conditions at prospective SCH sites. Report prepared by University of California Riverside for the California Department of Water Resources. May 9.

California Department of Fish and Game (DFG). 2011. Fish matrix: An analytical tool for selecting an aquatic community for proposed Species Conservation Habitat. Memorandum from Jack Crayon. April 22.

California Department of Water Resources (DWR) and California Department of Fish and Game (DFG). 2007. Salton Sea Ecosystem Restoration Program Final Programmatic Environmental Impact Report.

Cardno ENTRIX. 2010. Salton Sea species conservation habitat: Selenium treatment technologies. Final report prepared for the California Department of Water Resources. October.

Colorado River Basin Regional Water Quality Control Board (CRBRWQCB). 2006. Water quality control plan Colorado River Basin – Region 7.

Colorado River Basin Regional Water Quality Control Board (CRBRWQCB). 2010. Section 303(d) list for Colorado River Basin Region.

Holdren, C. Reclamation, unpublished data.

Johnson, P.I., R.M. Gersberg, M. Rigby, and S. Roy. 2009. The fate of selenium in the Imperial and Brawley constructed wetlands in the Imperial Valley (California). Ecological Engineering 35: 908-913.

Lin, Z., and N. Terry. 2003. Selenium removal by constructed wetlands: Quantitative importance of biological volatilization in the treatment of selenium-laden agricultural drainage water. *Environmental Science & Technology* 37:606–615.

Lorenzi, V. and D. Schlenk. In preparation. Draft report for Task Order #5 - Fish Tolerance. University of California Riverside.

MacDonald, D.D., C. F. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology* 39:20-31.

Miles A.K., M.A. Ricca, A. Meckstroth, and S.E. Spring. 2009. Salton Sea ecosystem monitoring project. U.S. Geological Survey Open File Report 2009-1976.

Ohlendorf, H.M., and G.H. Heinz. 2011. Selenium in birds. In *Environmental Contaminants in Biota: Interpreting Tissue Concentrations*, W.N. Beyer and J. Meador, eds. Boca Raton: CRC Press.

Orlando, J.L., K.L. Smalling, and K.M. Kuivila. 2008. Pesticides in water and suspended sediment of the Alamo and New Rivers, Imperial Valley/Salton Sea Basin, California, 2006–2007: U.S. Geological Survey Data Series 365.

Popma, T., and M. Masser. 1999. *Tilapia life history and biology*. Southern Regional Aquaculture Center Publication SRAC-283. March. Website (http://aqua.ucdavis.edu/DatabaseRoot/pdf/283FS.PDF) accessed March 29, 2011.

Presser, T.S. and S.N. Luoma. 2010. A methodology for ecosystem-scale modeling of selenium. *Integrated Environmental Assessment and Management* 6(4):685–710.

Sickman, J., J. Tobin, D. Schlenk, C. Amrhein, W. Walton, D. Bennett, and M. Anderson. 2011. Results from modeling of Se bioaccumulation potential in proposed Species Conservation Habitats of the Salton Sea. Report prepared for the California Department of Water Resources by University of California Riverside. February 9.

Wang, W., L. Delgado-Moreno, J. Conkle, and J. Gan. 2011. Survey of pesticide contamination in sediments at prospective SCH Sites. Draft report prepared for the Department of Water Resources. May 28.

J.8 Personal Communications

Amrhein, Chris. 2011. University of California Riverside. Personal communication with Ramona Swenson, Cardno ENTRIX, on April 22.

Orlando, James. 2011. U.S. Geological Survey. Personal communication with Ramona Swenson, Cardno ENTRIX, on June 2.